Nitrous Oxide Fluxes in Turfgrass: Effects of Nitrogen Fertilization Rates and Types

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Key Words: ammonium sulfate; urea; N_2O; urban lawns; dual-probe heat-pulse method;

Abbreviations: AS, High rate of ammonium sulfate fertilization; DOY, day of year; PVC, poly-vinyl chloride; UH, High rate of urea fertilization; UL, Low rate of urea fertilization.

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ABSTRACT

Urban ecosystems are rapidly expanding and their effects on atmospheric nitrous oxide (N_2O) inventories are unknown. Our objectives were to: 1) measure the magnitude, seasonal patterns, and annual emissions of N_2O in turfgrass; and 2) evaluate effects of fertilization with a high and low rate of urea N; 3) and urea and ammonium sulfate; on N_2O emissions in turfgrass. Nitrogen fertilizers were applied to turfgrass: 1) urea, high rate (UH; 250 kg N ha^{-1} yr^{-1}); 2) urea, low rate (UL; 50 kg N ha^{-1} yr^{-1}); and 3) ammonium sulfate, high rate (AS; 250 kg N ha^{-1} y^{-1}); high N rates were applied in five split applications. Soil fluxes of N_2O were measured weekly for one year using static surface chambers and analyzing N_2O by gas chromatography. Fluxes of N_2O ranged from -22 μg N_2O-N m^{-2} h^{-1} during winter to 407 μg N_2O-N m^{-2} h^{-1} after fall fertilization. Nitrogen fertilization increased N_2O emissions by up to 15 times within three days although the amount of increase differed after each fertilization; increases were greater when significant precipitation occurred within three days after fertilization. Cumulative annual emissions of N_2O-N were 1.65 kg ha^{-1} in UH, 1.60 kg ha^{-1} in AS, and 1.01 kg ha^{-1} in UL. Thus, annual N_2O emissions increased 63% in turfgrass fertilized at the high compared with the low rate of urea, but no significant effects were observed between the two fertilizer types. Results suggest that N-fertilization rates may be managed to mitigate N_2O emissions in turfgrass ecosystems.

Anthropogenic activities have contributed to an increase in concentrations of atmospheric N_2O, and agricultural soils may contribute more than 80% of anthropogenic N_2O emissions (Isermann 1994; Lal et al. 1999; Follet et al. 2000). A number of studies have determined that N_2O fluxes into the atmosphere are high in croplands on which N-fertilization and irrigation rates are also high (Goodroad and Keeney 1984; Sexstone et al. 1985; Mosier et al. 1986, 1991 & 1996; Bouwman 1996). However, urbanization in the United States and elsewhere is replacing significant tracts of land with turfgrass that were once occupied by natural or agricultural ecosystems (Heimlich and Reining 1989; Dillman et al. 1993; USGS 1999; Alig et al. 2004; Kaye et al. 2004), and urban expansion in the USA alone is projected to increase in area by 79% by the year 2025 (Alig et al. 2004). Because turfgrass is often irrigated and fertilized with N, urban areas may represent an unappreciated, but increasingly significant contributor to atmospheric N_2O.

The role of turfgrass ecosystems in atmospheric N_2O inventories is unknown (Bouwman et al. 1995; Kroeze 1999). Turfgrass has historically been excluded from regional N_2O inventories because of the assumption that urban land area is too small to make significant contributions. However, recent studies have indicated that 16 to 20 million ha of urbanized land, or up to 10% of the land area in some regions in the United States, are covered with turfgrasses (e.g., golf courses, sports fields, parks, home lawns, etc.) (Welterlen 1997; NASS 2004); this represents an area three times larger than any irrigated crop (Milesi et al. 2005).

Kaye et al. (2004) determined that turfgrass emitted 10 times more N_2O to the atmosphere than did native grassland and wheat-fallow soils in northeastern Colorado, USA. Those authors also estimated that turfgrass which covered 6.4% of the land area in their 1578 km^2 study contributed as much as 30% of the N_2O emissions. Research is needed to quantify

N_2O fluxes in turfgrasses including under various forms of land management (e.g., fertilization, irrigation, turfgrass species). Obtaining these data may be a first step in the development of best management practices in turfgrass that may mitigate the greenhouse effect by reducing effluxes of N_2O, which has about 300 times the warming power of CO_2 (IPCC 2001).

Fluxes of as much as 7,528 μg N_2O-N m^{-2} h^{-1} have been reported in perennial ryegrass (*Lolium perenne* L.) after N fertilization (Ryden 1981; Maggiotto et al. 2000). Maggiotto et al. (2000) determined that fertilizer type had significant effects on N_2O fluxes, with urea-based fertilizers resulting in lower N_2O emissions than other fertilizer types. Conversely, fertilizer type had no impact on N_2O emissions from a sward of Kentucky bluegrass (*Poa pratensis* L.)(Bergstrom et al. 2001). Others have reported that soil temperatures of 30^{o}C or higher coupled with saturated soil conditions increased denitrification rates in Kentucky bluegrass sod (Mancino et al. 1988); greater denitrification rates typically result in higher N_2O emissions (Firestone and Davidson 1989). Irrigation with as little as 5 mm of water increased emissions of N_2O markedly in a closely mown, mixed-grass sward of ryegrass and paspalum (*Paspalum dilatatum* Poir.)(Denmead et al. 1979). In another study, annual emissions of N_2O-N were estimated at 2.4 kg ha^{-1} from Kentucky bluegrass (Kaye et al. 2004).

Fluxes of N_2O from the soil surface originate primarily from the processes of nitrification and denitrification in the soil (Fig. 1) (Firestone and Davidson 1989). During nitrification, aerobic microbial populations convert ammonium (NH_4^+) to nitrate (NO_3^-); both NH_4^+ and NO_3^- are readily accessible by plant roots. Thereafter, NO_3^- can be converted to N_2 by anaerobic microbial populations via denitrification.

In turfgrass, the amount of N_2O evolution in the soil is affected by a number of cultural practices that closely regulate nitrification and denitrification (Fig. 1). For example, ammonium-

based fertilizers (e.g., ammonium sulfate) add ammonium (NH_4^+) directly to the soil and urea fertilizers are quickly converted to NH_4^+ in the soil, which is facilitated by the addition of water (e.g., from irrigation or precipitation). Nitrogen fertilization enhances nitrification, which requires NH_4^+ to proceed. Nitrification also requires adequate amounts of O_2 along with the presence of nitrifying microorganisms. Nitrification rates will decrease as soil water content approaches saturation (e.g., from irrigation or precipitation) because O_2 availability becomes limiting. Conversely, extremely dry soils may also slow nitrification if lack of moisture restricts microbial activity.

The general requirements for denitrification to occur include the restriction of O_2, the availability of N oxides, NO_3^-, NO_2^-, NO, or N_2O, an available organic carbon (C) source, and the presence of denitrifying microorganisms; the restriction of O_2 availability is likely the dominant factor limiting denitrification in soils followed by NO_3^- availability and then available organic C (Myrold 2004). Thus, the effects of various environmental controllers of denitrification and nitrification are somewhat hierarchal, ranging from more to less important (Firestone and Davidson 1989; Myrold 2004). Nitrogen fertilization and irrigation or precipitation affect both nitrification and denitrification and hence, N_2O fluxes, primarily by affecting proximal factors in the hierarchy (i.e., the amount of NH_4^+, NO_3^- and O_2 availability in the soil).

Other factors not discussed here may also affect nitrification and denitrification rates (e.g., physical properties of soils, temperature, pH). A comprehensive discussion of these factors was not included for the sake of brevity and because this study focused on the effects of N fertilization on N_2O fluxes. More complete descriptions of these factors and how they impact

nitrification and denitrification are available from a number of sources (Firestone and Davidson 1989; Myrold 2004).

In this study we measured N_2O emissions from turfgrass fertilized with two rates of urea and one rate of ammonium sulfate. The specific objectives of this study were to determine: 1) the magnitude, seasonal patterns, and cumulative annual emissions of N_2O_N fluxes in a turfgrass ecosystem; 2) the effects of a high and a low N-fertilization rate of urea on N_2O fluxes; and 3) the effects of two N-fertilizer types, urea and ammonium sulfate, on N_2O fluxes.

MATERIALS AND METHODS

The field study was conducted from 3 October (DOY 276) 2003 to 5 October (DOY 279) 2004 at the Rocky Ford Turfgrass Research Center near Manhattan, Kansas (Rocky Ford; 39.12^{o}N, 96.35^{o}W). The soil at the site was a Chase silt loam (fine, montmorillonitic, mesic, Aquic, Argiudolls). Mean annual air temperature was 11.6^{o}C, which was 1.1^{o}C below the 30-yr mean; precipitation was 887 mm, or 3 mm above normal. The summer months were substantially wetter and cooler than normal; precipitation between June and August 2004 was 468 mm, which was 148 mm above normal; the average temperature was 23.2^{o}C, or 2.1^{o}C below normal for the same period.

Fertilizer rate and type

Twenty four plots (2 x 2 m each) were established in September 2003 in an existing sward of perennial ryegrass; plots were separated by a minimum of 2 m. Three fertilizer treatments, replicated 8 times each, were applied to plots in a randomized block design. The treatments represented typical N fertilizer amounts and types in turfgrass and included: 1) Urea

N at a high rate (UH; 250 kg N ha^{-1} yr^{-1}); 2) Urea N at a low rate (UL; 50 kg N ha^{-1} yr^{-1}); and 3) Ammonium sulfate N at a high rate (AS; 250 kg N ha^{-1} y^{-1}). The high N rate is representative of intensively managed turfgrass such as in golf courses and the low rate a minimally maintained homeowner lawn. In UH and AS, urea and ammonium sulfate were applied in split applications of: 75 kg N ha^{-1} on DOY 279, 2003; 50 kg N ha^{-1} on DOY 318, 2003, DOY 86, 2004, and DOY 128, 2004; and 25 kg N ha^{-1} on DOY 191, 2004. In UL, 50 kg N ha^{-1} was applied on DOY 279, 2003.

After fertilizations, plots were irrigated with about 15 mm of water to incorporate fertilizer into the soil and reduce ammonia volatilization (Bowman et al. 1987). Plots were irrigated one to three times weekly or as needed to prevent drought stress. Irrigation applications were measured with in-line flow meters (Model FTB6205, Omega Engineering, Stamford, CT). Plots were mowed once or twice weekly as needed at a height of 7.5 cm with a walk-behind rotary mower.

The plot area was treated with herbicides on DOY 295, 2003 and DOY 148, 2004 for broadleaf control (carfentrazone-ethyl; 2,4-D ethylhexyl ester; Mecoprop-p acid; Dicamba, acid; 4.68 L ha^{-1}) and on DOY 148, 2004 for control of grassy weeds (dithiopyr; 4.68 L ha^{-1}). Fungicide (chlorothalonil; 14.00 L ha^{-1}) was applied on DOY 149, 2004 for dollarspot control and an insecticide (halofenozide; 112.3 kg ha^{-1}) was applied on DOY 191, 2004 for the control of white grubs.

Measurements of nitrous oxide fluxes

Soil-surface N_2O fluxes were measured by placing static, vented poly-vinyl chloride (PVC) chambers (7.5 cm high x 20 cm dia) using the method described by Hutchinson and

Mosier (1981; Mosier et al. 1991, 1997; Kaye et al. 2004). Using this method, temperatures inside the chambers were assumed to be equivalent with ambient air temperatures measured at a nearby weather station. Ambient temperatures inside the chambers were preserved by constructing them with white PVC and by placing highly reflective aluminum foil tape on the outside of the chambers to reflect radiation away from the chamber. In this study, gas was not mixed mechanically inside the chambers. In general, thermal buoyancy and/or external turbulence-induced pressure fluctuations support uniform mixing of the chamber head space and furthermore, mechanical gas mixing may be undesirable because it may artificially enhance gas exchange rates (Hutchinson and Livingston 2002).

Permanent PVC collars were placed randomly at one location in each plot and were driven approximately 8 cm into the soil. On measurement days, chambers were installed on the collars and gas samples from inside the chambers were removed with 12-ml polypropylene syringes fitted with nylon stopcocks at 0, 30, and 60 min. Gas samples were transported to the lab and analyzed by a gas chromatograph (Shimadzu GC14B, Shimadzu Scientific Instruments, Columbia, MD) equipped with a Porapak Q column (3.175×10^{-3} m diam. x 1 m, 80/100 mesh) and an electron capture detector. Gas samples were always analyzed on the same day as collected, and generally within 6 hours. Fluxes were calculated as described by Hutchinson and Mosier (1981) and Mosier et al. (1991).

The sampling frequency in the field was generally once weekly although measurements were more frequent after fertilizations to capture transient peaks in N_2O fluxes (Christensen 1983; Sexstone et al. 1985). Fluxes were not sampled during one 35-day period in late January through mid February (DOY 20-55, 2004), however, when snow cover prevented the measurement of fluxes. Gas samples were usually collected from 0700 to 1100 CST on each

measurement day to reduce the impact of diurnal variations in N_2O emissions (Blackmer et al. 1982; Christensen 1983; Goodroad and Keeney 1984).

Ancillary Measurements

Soil properties, soil temperature, volumetric soil water content, and periodically, NH_4^+ and NO_3^- and aboveground biomass production were measured to evaluate their relationships with N_2O emissions. Soil properties, including pH, texture and bulk density, were measured in the 0 to 10, 10 to 20, and 20 to 30 cm profiles (Table 1). Ammonium and nitrate concentrations in the 0-10 cm profile were measured intensely on 4 days during the growing season, concurrent with N_2O flux measurements (DOY 147, 174, 202, and 216, 2004). On each of the 4 days, soils from each of the 36 plots were sampled separately for NH_4^+ and NO_3^- concentration to provide rigorous, direct correlations with N_2O fluxes from each respective plot.

Climatological variables were measured at a weather station located at Rocky Ford Turfgrass Research Center. To determine irrigation requirements, evapotranspiration (ET) was calculated by using the Penman-Monteith equation (FAO-56; FAO 1998).

Soil water filled porosity (WFP) and temperature at 5 cm were measured automatically using the DPHP technique (Campbell et al. 1991; Tarara and Ham 1997; Song et al. 1998). Sensors were fabricated in the laboratory as described by Basinger et al. (2003) and Bremer (2003). Measurements of WFP were logged once daily at 0600 CST and soil temperatures were logged every 60 min. All data acquisition and control were accomplished with a micrologger and accessories (CR10x and one AM16/32, Campbell Scientific, Logan, UT). Bulk density of the soil at 5 cm was 1.35 g cm^{-3} (determined from volumetric samples 5.4 cm diam. x 3 cm); organic matter was 4% (Soil Testing Laboratory, Kansas State University).

Clippings were collected periodically from DOY 159 to 288, 2004 with a walk-behind rotary mower equipped with a modified collection bag that allowed for complete capture of clippings from each plot. Clipped biomass was determined gravimetrically after samples had been dried in a forced-air oven for 48 h at 65^{o}C.

Data analysis

Nitrous oxide flux data from each plot on each measurement day were evaluated to ensure fitness and flux rates were then calculated. Cumulative emissions of N_2O were estimated as the sum of the products of weekly flux rates and the number of days between samples. Cumulative values were first calculated for each plot, and treatment values were then obtained by averaging cumulative values over all plots within each respective treatment. If a measurement preceded or followed fertilization, which had large effects on N_2O emissions, the interpolation was truncated based on the timing of the fertilization.

Tests of differences among treatments in measurements of N_2O fluxes, gravimetric estimates of clipping biomass, and ammonium and nitrate concentrations in the soil were conducted by using the mixed linear model of SAS ($P<0.05$; SAS Institute 2003); fixed effects included fertilizer type and amount and random effects included block. Correlations between soil N and N_2O fluxes were conducted with the correlation procedure (Spearman) of SAS.

RESULTS AND DISCUSSION

Fluxes from fall 2003 through spring 2004

Fluxes of N_2O-N increased by five to 15 times in UH and AS plots within three days after fertilization in the fall of 2003 (Fig. 2) and were the highest observed in the year-long study; 36

mm of rainfall two days after fertilization likely contributed to the higher fluxes (Sexstone et al. 1985; Clayton et al. 1994). Peak fluxes were measured three days after fertilization at 407 μg N_2O-N m^{-2} h^{-1} in UH and were comparable to N_2O-N fluxes measured in fertilized crops and grasslands (Mosier et al. 1991; Clayton et al. 1994; Kaiser et al. 1998; Webb et al. 2004). Increases in N_2O fluxes are not unusual after fertilization in crops or grasslands (Mosier et al. 1991; Clayton et al. 1994; Webb et al. 2004) although no change or even negative fluxes have been observed after N fertilizations (Maggiotto et al. 2000; Webb et al. 2004).

Fluxes decreased rapidly after peaking and generally began to level off after about 1 to 2 weeks, which is similar to the duration of increases in N_2O emissions observed after N fertilization in other grassland or turf studies (Ryden 1981; Christensen 1983; Bergstrom et al. 2001). Fluxes were significantly higher in UH than in AS (DOY 280) and UL (DOY 283)(Fig. 2), and cumulative fluxes in the week following fertilization were significantly higher in UH than in UL (data not shown).

Later in the fall of 2003, fluxes again increased in UH and AS after N fertilization in those plots (Fig. 3A). Four days after fertilization (DOY 322), fluxes were significantly higher in UH and AS than in UL, and peaked at 68 and 55 μg N_2O-N m^{-2} h^{-1} in UH and AS, respectively.

With the exception of the increase after N fertilization, N_2O fluxes generally decreased among treatments during the fall and early winter of 2003 (DOY 301 to 357) following the trend of soil and air temperatures (Fig. 3B). On DOY 346, fluxes were negative for the first time and were the largest negative fluxes observed during the study; greatest negative fluxes were -22 μg N_2O-N m^{-2} h^{-1} in UH. Lower temperatures, moist soils (Figs. 3B and 3C), and low NO_3^- availability presumably resulted in the consumption of N_2O in the soil (and hence, from the atmosphere) by denitrifying bacteria (Firestone and Davidson 1989). Negative N_2O fluxes have

been observed in a number of ecosystems (Mosier et al. 1996; Teira-Esmatges et al. 1998; Webb et al. 2004), including in turfgrass (Ryden 1981). With few exceptions, fluxes were generally slightly negative or negligible during the winter until after the spring fertilization on DOY 86, 2004. Small winter fluxes in this study contrasts with fluxes in some native grasslands and fertilized crops in which 20 to 50% of annual N_2O emissions occurred during winter (Mosier et al. 1996; Kaiser et al. 1998).

On DOY 55, N_2O fluxes were positive in AS but differences were not statistically significant among treatments (Fig. 3). The higher average flux in AS on DOY 55 resulted from a high flux rate in only one plot, which indicates large spatial variability of N_2O emissions from the soil as reported by other researchers (Parkin 1987; Clayton et al. 1994; Bergstrom et al. 2001).

After N fertilization in UH and AS on DOY 86, 2004, N_2O fluxes increased among plots including in UL which were not fertilized (Fig 3A). The increase in UL was likely caused by an increase in soil water content from 38 mm of precipitation in the 2 days after fertilization (Fig. 3C)(Denmead et al. 1979; Sextone 1985). Precipitation also may have caused additional increases in N_2O fluxes in UH and AS and thus, amplified or confounded the effects of N fertilization (Clayton et al. 1994). Fluxes of N_2O were generally higher in fertilized than in unfertilized plots during a 25 day period after fertilization although fluxes in AS were significantly lower than in UH on 2 of the 5 measurement dates during that period. Significant differences remained among treatments on DOY 111, which was more than 3 weeks after N fertilization.

Fluxes during summer 2004

During the summer of 2004, N_2O fluxes again increased markedly after N fertilization on DOY 128 (Fig. 4A). Four days after fertilization, fluxes of N_2O had increased in all treatments including in UL, which was not fertilized at that time. As in the earlier situation (after N fertilization on DOY 86, Fig. 3), 24 mm of precipitation 3 d after fertilization (DOY 131) increased soil moisture (Fig. 4C) and thus, likely inflated fluxes in UL. In UH and AS, precipitation also may have amplified or confounded the effects of fertilization on N_2O fluxes. Soil temperatures also increased by nearly 5 ^{o}C between the pre- and post-fertilization measurements (Fig. 4B) and may have further inflated N_2O fluxes among treatments (Denmead et al. 1979; Christensen 1983; Goodroad and Keeney 1984).

Later, after fertilization of UH and AS on DOY 191, N_2O fluxes increased considerably in UH and AS but only slightly in UL. The slight increase in UL despite receiving no N fertilization was similar to a small increase observed in UL on DOY 318, 2003 (Fig. 3A), after fertilization of only UH and AS. On neither of these occasions was precipitation a factor. However, post-fertilization irrigations that were applied to all plots including UL may have contributed to the slight increases in N_2O emissions in UL (Denmead et al. 1979; Sextone et al. 1985).

Correlations were significant but weak between N_2O fluxes and soil NO_3^- ($r=0.52$; $P<.0001$) and non-significant between N_2O fluxes and NH_4^+ (data not shown). Low correlations between N_2O fluxes and soil NH_4^+ and NO_3^- have been reported by others and some have suggested that N_2O emissions may be more related to soil N turnover rates than to mineral N pool size (Schimel et al. 1989; Davidson and Hackler 1994; Matson and Vitousek 1990). Mosier et al. (1996) found low correlations between N_2O fluxes and soil N concentrations and reported

that because N_2O production is determined by complex interactions among soil N concentration, water, temperature, etc., direct correlations between N_2O fluxes and any one of these variables was typically low.

Aboveground biomass production (clippings), which indicates the availability of organic C that is an important controller of denitrification, also was not a strong indicator of N_2O fluxes (data not shown). Clippings were significantly less in UL than in UH and AS on all measurement dates except for the first (DOY 148) and yet N_2O fluxes were only significantly lower in UL in three out of 15 days when fluxes were measured between DOY 118 and 244.

Annual fluxes of nitrous oxide and comparisons to other ecosystems

Cumulative annual emissions of N_2O-N were 58 and 63% greater in UH and AS, respectively, than in UL (Fig. 5). Cumulative emissions of N_2O-N were 1.65 kg ha^{-1} yr^{-1} in UH, 1.60 kg ha^{-1} yr^{-1} in AS, and 1.01 kg ha^{-1} yr^{-1} in UL; annual N_2O emissions were significantly greater from UH than UL. Thus, although transient differences between weekly fluxes in UH and AS were observed during the study, differences in annual N_2O emissions between UH and AS were not significant indicating fertilizer type had no effect on N_2O emissions in this study.

Over the year, an equivalent of approximately 0.65% in UH and AS and 2.02% in UL of annually applied N fertilizer was emitted into the atmosphere during the 1-yr study. These results are similar to the 2.2% of added N fertilizer emitted into the atmosphere as reported in the only other experiment where annual N_2O emissions were measured in turfgrass ecosystems (Kaye et al. 2004). In the latter experiment, annual N_2O emissions in turfgrass were comparable to emissions in irrigated corn, and were 10 times greater than in native grassland and wheat-fallow soils.

Cumulative emissions of N_2O were rapid during high fluxes after the fall fertilization in 2003 (Fig. 5). However, during the winter when soils were cool and wet and N_2O fluxes were slightly negative (Fig. 3A), cumulative N_2O emissions decreased as soil microbes absorbed atmospheric N_2O during denitrification. Reduction of N_2O to N_2 is possible when the production of N_2O is limited by a low supply of NO_3^-. As a result, denitrifiers use N_2O as the substrate. (Dr. Charles Rice, personal communication). During the winter period, cumulative fluxes of N_2O decreased by 0.11 to 0.20 kg ha^{-1} among treatments compared with cumulative emissions during October and November. Thereafter, during the spring and summer months, cumulative N_2O emissions increased because fluxes were generally positive.

In general, annual emissions of N_2O in this study were similar to or lower than N_2O fluxes from other urban or agricultural grasslands. For example, annual emissions from a Kentucky bluegrass lawn were 2.4 kg N_2O-N ha^{-1} yr^{-1} during a 1-yr study (Kaye et al. 2004), and fluxes from native grassland and wheat ecosystems in the United States ranged from 0.24 to 2.26 kg N_2O-N ha^{-1} yr^{-1} (Mosier et al. 1991, 1996). Dobbie et al. (1999) reported annual emissions in cut ryegrass in Scotland that ranged from 1.9 to 18.4 kg N_2O-N ha^{-1} yr^{-1} over a 3-year period. In their study, significant interannual variation in N_2O fluxes resulted from the same field; because the field was not irrigated, wet and dry periods were responsible for most of the variation. Emissions of N_2O ranged from 1.5 to 7.8 kg N_2O-N ha^{-1} yr^{-1} in grass, barley, and fallow in agricultural mineral soils in the boreal region (Syvasalo et al. 2004) and were 11.2 kg N_2O-N ha^{-1} yr^{-1} in a fertilized grassland in Germany where N turnover rates were rapid (Tilsner et al. 2003). Annual emissions as high as 56 kg N_2O-N ha^{-1} were reported from rye cropland in peat soils in Germany (Flessa et al. 1998) and from long-term, manured agricultural soils in Canada (Chang et al. 1998) where pH was low and groundwater levels were high.

Estimates of annual N_2O emissions in this study also were comparable to annual N_2O emissions from a number of cereal crops, sugarbeets (*Beta vulgaris* L.), potatoes (*Solanum tuberosum*), and linseed (*Linum usitatissimum*), which never exceeded 2 kg N_2O-N ha^{-1} yr^{-1} in a 6-yr study in the United Kingdom (Webb et al. 2004). However, emissions in the current study were lower than fluxes in rice, cotton, and vegetable croplands in the United States (6.5-8.4 kg N_2O-N ha^{-1} yr^{-1}; Mummey et al. 1998), from maize in Germany (3.9 to 8.7 kg N_2O-N ha^{-1} yr^{-1}; Sehy et al. 2003), and from onion cropland in Japan (15.6 kg N_2O-N ha^{-1} during the growing season; Kusa et al. 2002) where soil water content was high and N-fertilizer was applied at a higher rate than this study. Annual emissions in the current study were also lower than in fallow agricultural organic soils in the boreal region (23.5 kg N_2O-N ha^{-1} yr^{-1}; Maljanen et al. 2004); high fluxes in the latter study were from wet, N-fertilized, bare soils where the absence of plants resulted in higher soil NO_3^- concentrations.

CONCLUSIONS

Nitrogen fertilization increased N_2O emissions by as much as 15 times within three days although the amount of increase varied after each fertilization; increases tended to be greater when significant precipitation occurred within two or three days of N fertilization. Fluxes of N_2O ranged from -22 μg N_2O-N m^{-2} h^{-1} during winter to 407 μg N_2O-N m^{-2} h^{-1} after fall fertilization. Cumulative annual emissions of N_2O-N were 1.65 kg ha^{-1} in UH, 1.60 kg ha^{-1} in AS, and 1.01 kg ha^{-1} in UL. Thus, N fertilization at the high rate of urea increased annual N_2O emissions in turfgrass by 63% compared with N fertilization at the low rate. Emissions of N_2O were similar, however, between turfgrasses fertilized with the same rate of ammonium sulfate and urea N. In general, N_2O fluxes were higher for about two or three weeks after N fertilization before

decreasing to background levels. Further long-term research is necessary to measure N_2O emissions in turfgrasses, including in different geographical areas with different turfgrass species or considering interannual variations in climatic conditions at individual sites, and to determine best management practices that may reduce N_2O emissions in turfgrasses.

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Table 1. Soil site properties in study area.

Depth (cm)	pH	Bulk Density (g cm^{-3})	Sand, %	Clay, %	Silt, %
0-10	6.6	1.35	14	29	57
10-20	7.2	1.42	14	31	55
20-30	7.2	1.47	14	34	52

List of Figures

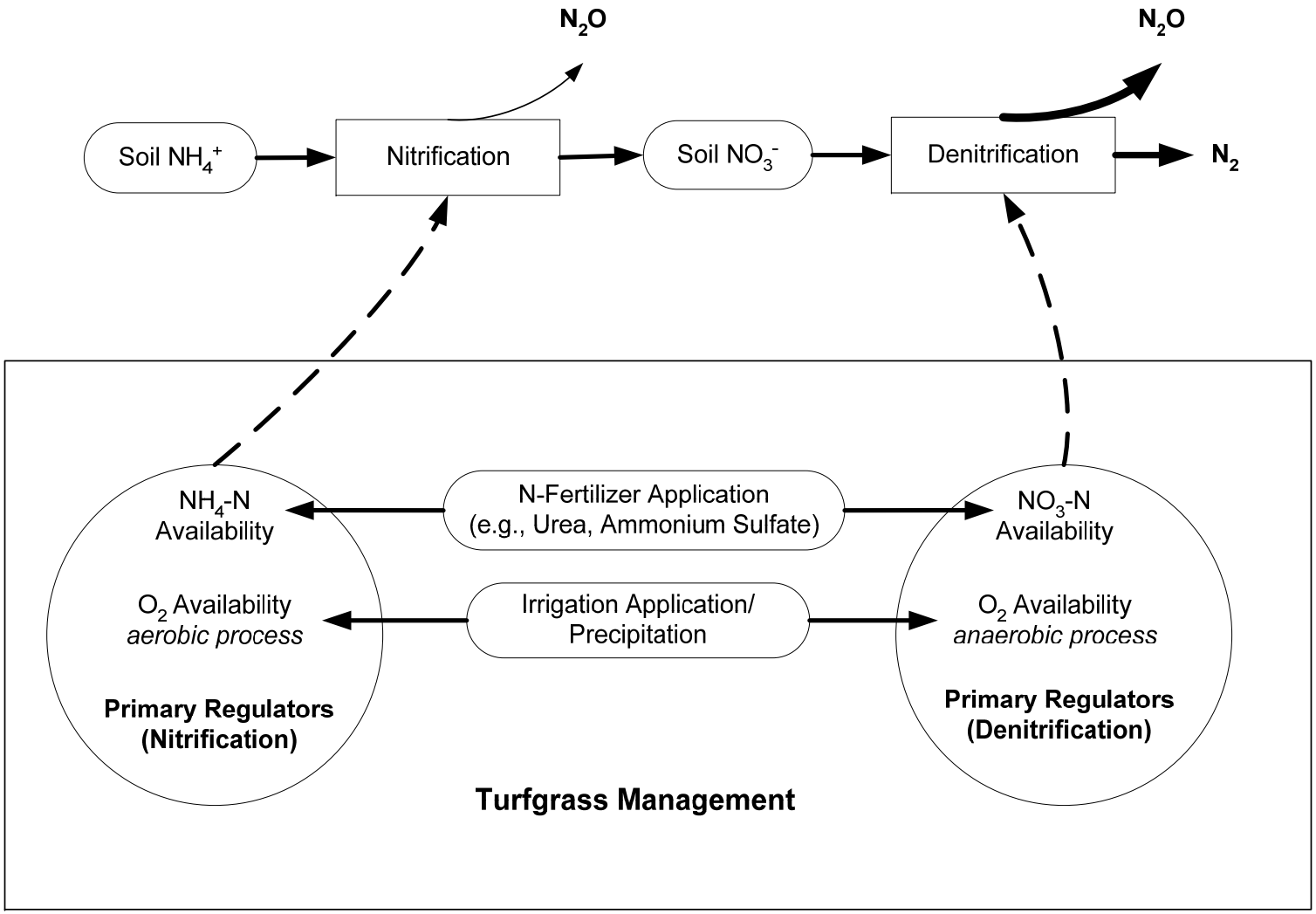
N2O
N2O
Soil NH4+
Nitrification
Soil NO3-
Denitrification
N2
NH4-N
Availability
N-Fertilizer Application
(e.g., Urea, Ammonium Sulfate)
NO3-N
Availability
O2 Availability
aerobic process
Irrigation Application/
Precipitation
O2 Availability
anaerobic process
Primary Regulators
(Nitrification)
Primary Regulators
(Denitrification)
Turfgrass Management

Figure 1

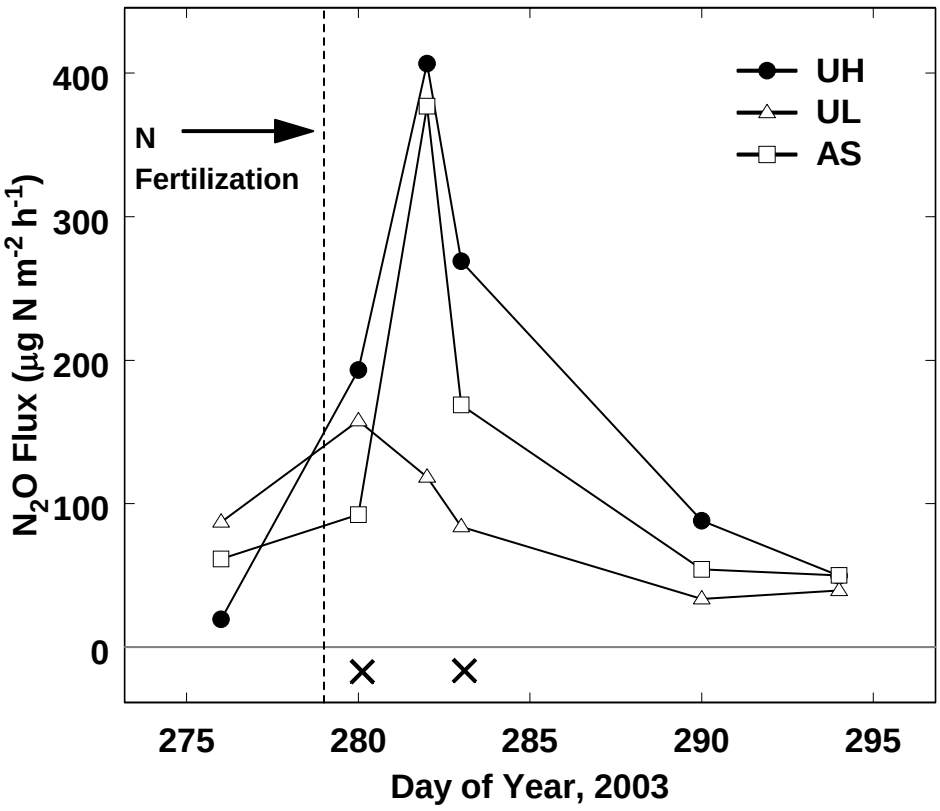
400
300
200
100
0
N
Fertilization
UH
UL
AS
N2O Flux (μg N m-2 h-1)
275
280
285
290
295
Day of Year, 2003

Figure 2

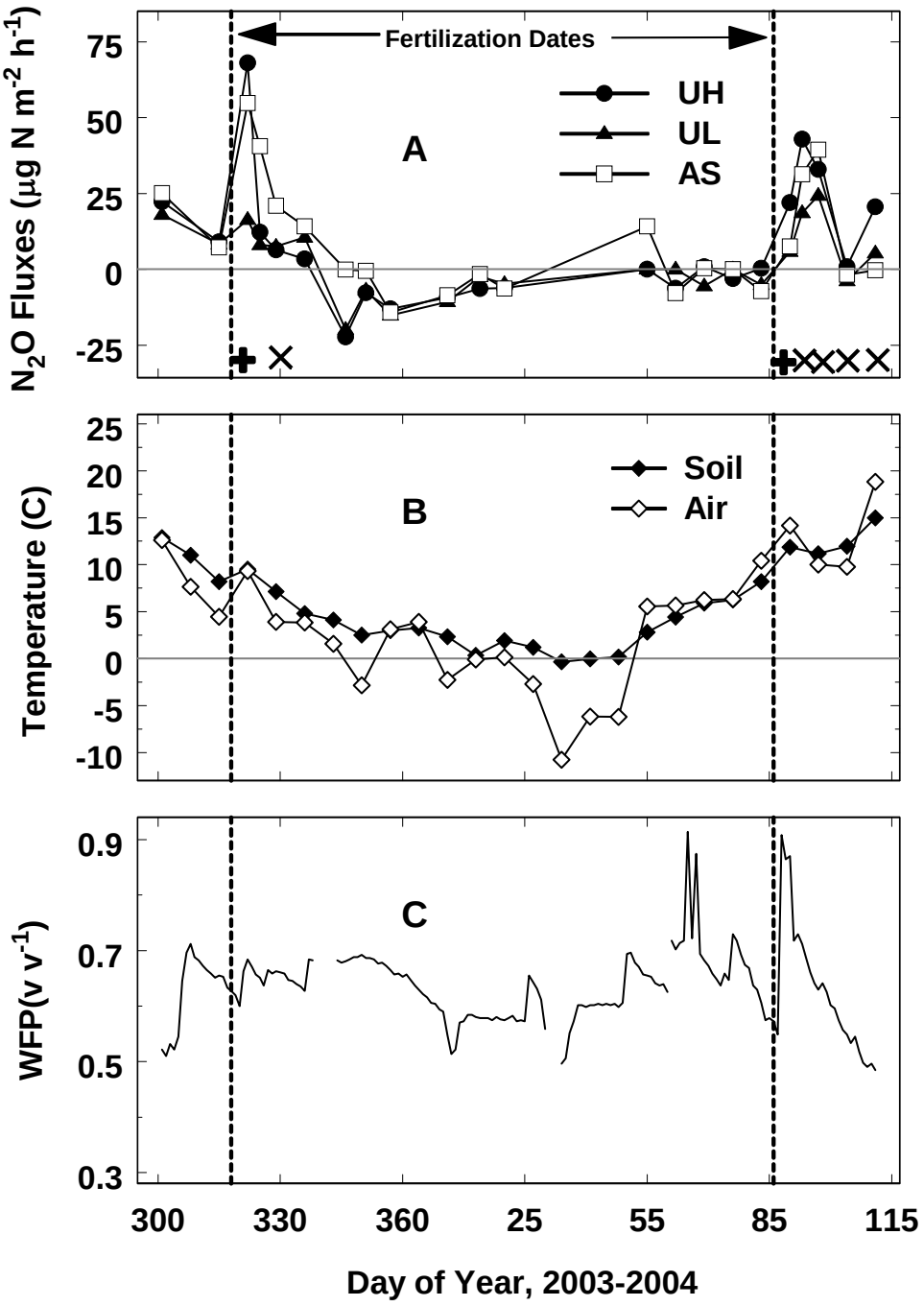
Fertilization Dates
UH
UL
AS
A
B
Soil
Air
C
N2O Fluxes (μg N m-2 h-1)
Temperature (C)
WFP(v v-1)
75
50
25
0
-25
25
20
15
10
5
0
-5
-10
0.9
0.7
0.5
0.3
300
330
360
25
55
85
115
Day of Year, 2003-2004

Figure 3

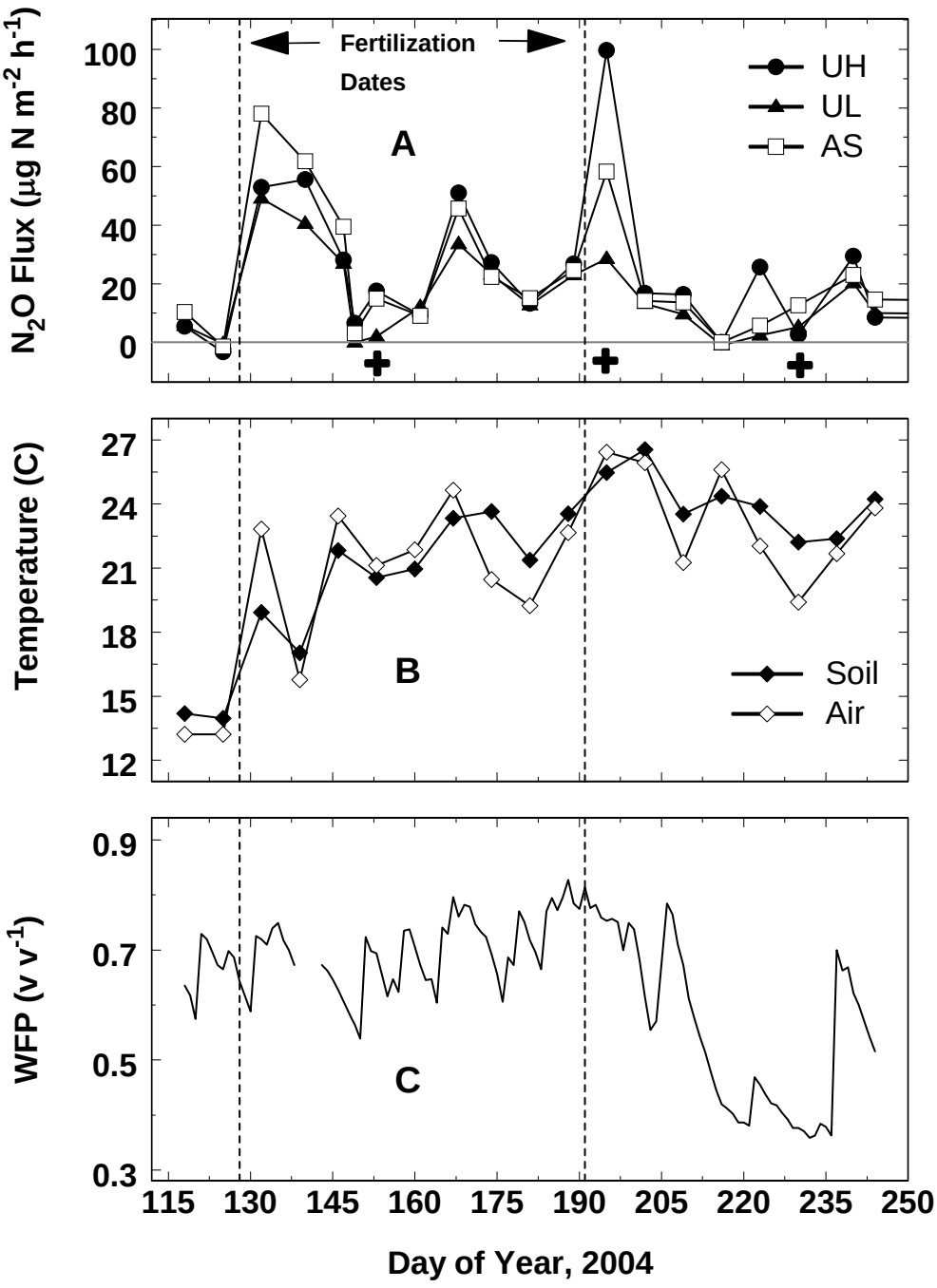
Fertilization
Dates
A
UH
UL
AS
N_2O Flux (μg N m^{-2} h^{-1})
100
80
60
40
20
0
B
Soil
Air
Temperature (C)
27
24
21
18
15
12
C
WFP (v v^{-1})
0.9
0.7
0.5
0.3
115
130
145
160
175
190
205
220
235
250
Day of Year, 2004

Figure 4

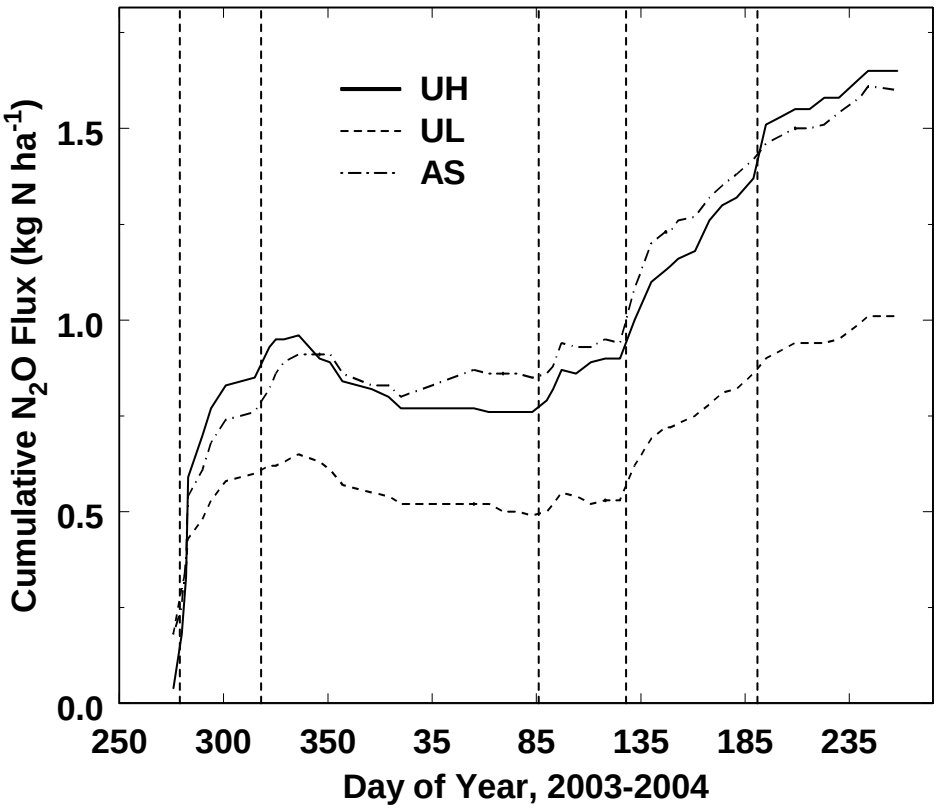
UH
UL
AS
Cumulative N_2O Flux (kg N ha^{-1})
0.0
0.5
1.0
1.5
250
300
350
35
85
135
185
235
Day of Year, 2003-2004

Figure 5